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Long Island Sound

Prospects for the Urban Sea

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transmission lines while LILCO's power plants were sold to private companies (NYT, Feb. 25, 1998, B4). To ensure an adequate supply of electricity for the Island's growing population, LIPA erected several small conventional power plants and, as back-up, the Trans-Energie underwater cable built by Hydro-Quebec was laid between Connecticut and Shoreham in 2002 (NYT, Jan. 13, 2002, LI2).

The value of a cross-Sound cable had been demonstrated in 1996 when power from Long Island was transmitted through an earlier pipeline to supplement Connecticut's supply during a period when all three of the reactors at the Millstone Nuclear Power Plant had been taken off line because of safety considerations. Interestingly, when Northeast Utilities began building the Millstone facility in Waterford (near New London) in 1966, there was a noticeable absence of opposition. In time, some residents expressed concern when a third reactor was constructed but by and large the tax revenue pouring into the municipality from Northeast Utilities was viewed as a fair trade-off. By the early twenty-first century, however, opposition to the Millstone plants surfaced on eastern Long Island. Given its proximity to coastal Connecticut and the impossibility of evacuation in the event of a serious malfunction at the Waterford facility, the East End would be a dead end if a radioactive plume drifted across the Sound (NYT, March 2, 1999, B7).

Safety concerns notwithstanding, aging nuclear power plants are permitted to remain online because of the demand for the energy they produce. As the twenty-first century progresses, it is conceivable that more such plants will be constructed along the Sound. Of course there will be mass demonstrations aimed at preventing this from happening, especially in view of leakage from Japanese nuclear plants in the wake of the 2011 tsunami, just as there was considerable opposition to the plan to build a liquid natural gas terminal in LIS. Proposed by Broadwater Energy, a joint venture of TransCanada Corporation and Shell Oil, the seven-story-high floating terminal, with a length equivalent to three football fields, would have served as a processing facility where the liquefied product, brought in by ship, would have been turned into gas and piped to New York and Connecticut. Opposition by environmentalists, fishermen, and residents of the LIS area caused New York State to reject Broadwater Energy's proposal in 2008 (Environmental News Service, April 11, 2008). In April 2009, the US Commerce Department announced that it would not grant federal permits for the project (NYT, April 16, 2009, 28).

1.6.2 Dredging and Dumping

Even before the Broadwater decision, there was speculation about alternative sources of energy, including wind turbines proposed for the waters off Plum Island. Whether this or as yet unheard of methods of providing power to the LIS region, it would seem that nuclear power plants will continue to provide some of the area's energy, despite decades-old concerns about this method of generating electricity. Back in the 1970s, an in-depth study of the Sound by the New England River Basins Commission (NERBC) (NERBC 1975), a federal agency, noted that thermal pollution from both conventional and nuclear power plants was one of six major ecological threats to LIS. The other challenges were radioactive pollution from nuclear power plants, oil leaks and spills, sewage pollution, destruction of wetlands, and dredging and dumping. In one sense, all of these challenges related to increased population. More people needed more power, and the vessels delivering oil to electrical generating facilities or to home heating oil tank farms had occasional accidents. In 1969 and again in 1972, tankers hit Barlett's Reef in the eastern Sound. The 1972 spill covered nearly 50 square miles and oil washed ashore at Niantic where it coated waterfowl. To protect shellfish beds in the Niantic River, the State of Connecticut used containment booms to seal off the river from the Sound (NYT, March 23, 1972, 53). That same year, a brand new oil tanker split in two in Port Jefferson Harbor. Fortunately the oil had already been unloaded, but the accident made people wonder whether a proposed dredging project to deepen the harbor to permit larger tankers to access the LILCO power plant and the Consolidated Petroleum Company should go forward (NYT, Jan. 11, 1972, 1). In the last four decades, dredging has become controversial. Yet, there is an ongoing need to dredge approaches to power plants, harbors, and marinas accommodating pleasure craft whose numbers have continued to grow, recessions and fuel price spikes notwithstanding. The permitting process for these jobs is often lengthy and arduous because a host of potential environmental impacts must be taken into account, among them spawning and migration of different species of fish and nesting habits of shorebirds. The end result is increasingly narrow windows for dredging. An occasional frigid winter resulting in frozen inlets can further reduce the permissible time for dredging. Then there is the challenge of what to do with the dredged material.

When Thames River dredging was done in the mid-1990s to enable *SEAWOLF* submarines to access their base in Groton, the dredged material was dumped in LIS. Since the Sound is the only non-ocean governed by the 1972 Marine Protection, Research and Sanctuaries Act's strict testing regulations, New York State initiated legal action to halt the dumping because of the possible effect upon lobsters from toxins in the dredged material. Although the dumping was allowed to continue, stricter adherence to the regulations was required. This did not end the battle over dumping, however. For the next decade, Connecticut continued to assert that from an economic standpoint disposing of dredged materials in the Sound was the only way to keep the state's harbors open to commerce. All the while New York insisted that alternatives had to be sought. In 2007, an agreement brokered by the federal government made 2014 the deadline for implementing a new dredged material management plan. Without a satisfactory plan, dumping of dredged material in the Sound will cease that year (NYT, Jan. 13, 2008, LI14).

1.6.3 The LIS Studies

Dredging and dumping, as well as the other issues highlighted in the NERBC study of the 1970s, were revisited when the federal government and the states of may experience tidal current exchange, i.e., seasonal disturbance from storms or T, but most disturbances occur with periodicities measured in months or years. Most subtidal habitats in LIS are well protected from frequent natural disturbance with the exception of areas scoured by tidal currents near The Race or on shallow ridges or shoals (e.g., Stratford Shoal, Long Sand Shoal). However, the tidal exchange of sediment across the sediment–water interface can be quite substantial in many of these subtidal habitats.

The seasonal influx of suspended sediment into the Sound has been estimated to be 9.3×10^8 kg year^{-1} (Farrow et al. 1986; Rhoads 1994), equivalent to an annual sediment mass accumulation rate of 0.05 g cm^{-2} year^{-1} over 1792 km^2 of muddy sediment (Bokuniewicz and Gordon 1980). This estimate compares with long-term sedimentation rates determined by ^{210}Pb profiles from the center of LIS of 0.05 g cm^{-2} year^{-1} and radiocarbon dating results of 0.077 g cm^{-2} year^{-1} (Benoit et al. 1979). Varekamp et al. (2010) measured mass accumulation rates in cores from LIS (using ^{210}Pb, ^{137}Cs, and ^{14}C) of 0.01–0.05 g cm^{-2} year^{-1} before the 1800s; rates increased to approximately 0.25 g cm^{-2} year^{-1} in the mid-1800s. The deposition rate may be higher (0.1 g cm^{-2} year^{-1}) in the western basin (Bokuniewicz 1988), but Varekamp et al. (2010) did not confirm this observation. In addition to the influx of suspended sediment, the central and western basins are subject to tidal resuspension and deposition of large volumes of sediment (Rhoads et al. 1984; Knebel and Poppe 2000). Most of this resuspended sediment is trapped below the pycnocline in summer and creates a transitory near-bottom turbidity zone (Rhoads et al. 1984). Average near-bottom turbidity values of 5 mg L^{-1} result in an estimate of 2.5×10^8 kg or 27 % of the annual supply in suspension with higher values in spring and early summer (Bokuniewicz 1988; Rhoads 1994). The flux of sediment due to suspension and redeposition appears to be much higher than the net long-term sedimentation (Rhoads 1994). Using McCall's sediment trap data, Rhoads calculated that 1×10^{12} kg of fine sediment was resuspended annually or the equivalent of 1,000 times the long-term sedimentation rate (McCall 1977; Rhoads 1994). These measured and estimated rates suggest that the benthic environments in the central and western basins experience very high fluxes of fine-grained sediments and that very little of the net influx of sediment is removed from the resuspension cycle (Rhoads 1994). The benthic community in these environments is therefore exposed and presumably adapted to relatively high exchange of sediment across the sediment–water interface, despite the apparently protected conditions (McCall 1977). During certain storm events, the resuspension levels may be much higher, but there is little evidence that storm events are a major source of disturbance except in shallow nearshore habitats.

Dredging and Dredged Material Disposal

Dredging affects channel floor habitats, with minimal loss of suspended sediments to surrounding habitats (Bohlen et al. 1979; Wilber et al. 2007). Material removed during dredging is frequently placed on the sea floor in open water habitats with immediate, short-term effects on the benthic community (Germano et al. 1994; Bolam and Rees 2003). The annual average dry weight of dredged material placed in LIS has been estimated at 4.1×10^8 kg year^{-1} or less than half of the annual sedimentation rate (Rhoads 1994). Of this amount, dispersal losses from passage through the water column and resuspension after placement are estimated at 6 %, or about 3 % of the annual non-disposal sediment input (Rhoads 1994).

Historically, dredged material disposal in the Sound occurred at many sites located just outside harbors adjacent to the recognized channel (Fredette et al. 1993). As a result, some benthic habitats still retain traces of placement of isolated piles of harbor sediment (Poppe et al. 2001, 2010; ENSR 2007).

Disposal is now confined to designated areas between 3.43 and 6.86 km^2 (1–2 nautical miles^2 in deeper areas of LIS (Fredette et al. 1993). Disposal of dredged sediments is permitted for materials that are determined to be suitable for unconfined open water disposal based on biological based testing protocols (Fredette and French 2004; EPA/USACE 1991). Disposal material typically consists of seasonal placement of 10–500,000 cubic yards (7.65–382,300 m^3) of harbor sediments at buoys by releasing the material from a split-hull barge at the surface. Each barge contains between 382 and 2,294 m^3 (500–3,000 yd^3 of water-laden sediment. Recent sea floor imaging studies, experimental placement, and laboratory experiments have clarified the physical processes involved in placement of dredged material in open water (Fig. 6.33; Valente et al. 2012; ENSR 2007). The sediment released from the bottom of the barge falls rapidly to the sea floor, entraining some water but retaining a coherent mass until it contacts the sea floor. Upon contact, the vertical force of the bolus of water-entrained sediment is transferred to lateral forces and rapidly spreads in a circular pattern to form a low mound or crater shape on the sea floor (Fig. 6.34). Depending on the water depth, barge volume, water content of the dredged material, and the seafloor surface, the mound or crater is between 50 and 300 m in diameter. This process of sediment placement creates a disturbed sediment surface that consists of a coherent layer of dredged material 10–200 cm thick in the center of the mound, thinning to mixed layers of ambient sediment and fresh dredged material. At the outer margin of the placement feature, layers 1–2 mm thick of fresh dredged material can be detected with sediment profile imaging techniques (Germano et al. 2011). This process of placement is usually repeated 10–250 times at a single disposal buoy during a disposal season (October–May), resulting in a shallow mound 1–5 m thick and 200–1,500 m in diameter (Figs. 6.33 and 6.34; Valente et al. 2012; ENSR 2007).

After placement of sediments on the sea floor, the resulting mound remolds over a period of months due to consolidation (Silva et al. 1994; Poindexter-Rollings 1990), bioturbation, erosion, and deposition from near-bottom flow processes (Rhoads 1994). As a result, the volume of the mound will decrease and the surface will smooth and begin to converge with conditions of adjacent ambient sediments. The degree of convergence will depend on water depth, physical conditions of the placement site, and the differential between placed and ambient sediments (Rhoads 1994). Management of placement sites for containment of dredged material dictates creation of stable mounds in depositional environments (Fredette and French 2004).

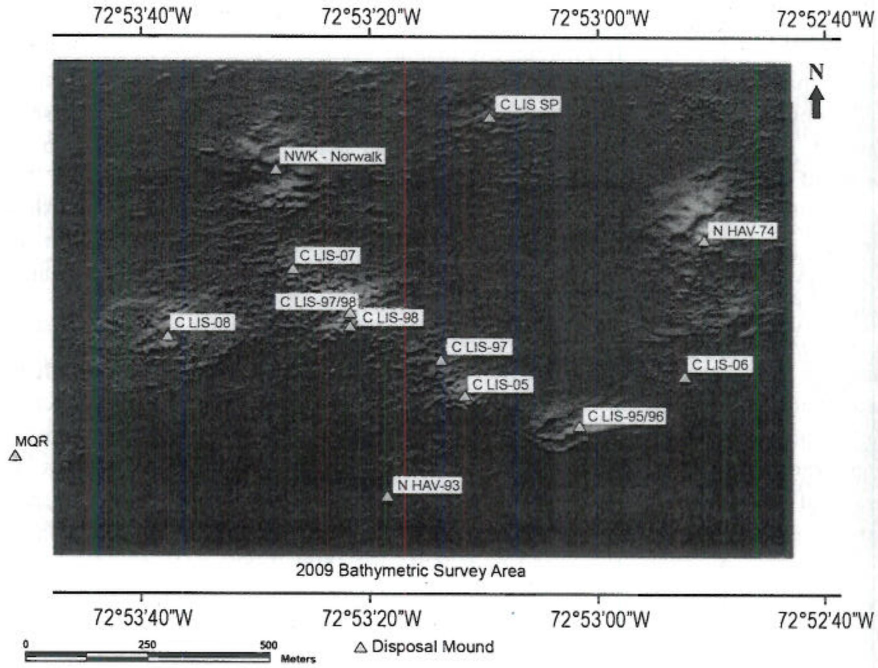

Fig. 6.33 Hillshaded multibeam bathymetry of a portion of the Central Long Island Sound Disposal Site surveyed in 2009. Individual mounds (e.g., CLIS-05) represent accumulated disposal of dredged material for one or more disposal seasons (October–May). Relief is exaggerated to highlight low relief features that result from impact of dredged material on the seafloor (from Valente et al. 2012)

Mounds containing dredged material unsuitable for unconfined placement due to elevated levels of contaminants are engineered to meet regulatory guidelines under which the contaminants are made inaccessible (Fredette and French 2004; EPA/USACE 1991). Studies of the stability of these engineered or "capped" mounds (mounds constructed to isolate the unsuitable material beneath a sediment "cap"; Palermo 1991) have demonstrated that pore water exchange of contaminants with the overlying ambient water is well below background levels (Bokuniewicz 1989; Murray et al. 1994). Longer term studies of mounds have demonstrated that once the rapid consolidation phase has been completed (ca. 1 year), the surface layers of sediment (10–30 cm) are the only horizons available to interact with biological resources (Fredette et al. 1992; Murray et al. 1994).

The consequence to benthic habitats of the placement of dredged material has been studied in LIS for over 40 years (Fredette and French 2004; Valente 2004). The nature of the impacts can vary depending on the composition of the dredged material and the habitat at the disposal site (Bolam et al. 2006). A structured monitoring approach of disposal impacts has been utilized in LIS since 1977 as part of the Disposal Area Monitoring System, or DAMOS (Germano et al. 1994; Fredette and French 2004). Based on the results of DAMOS studies and the understanding

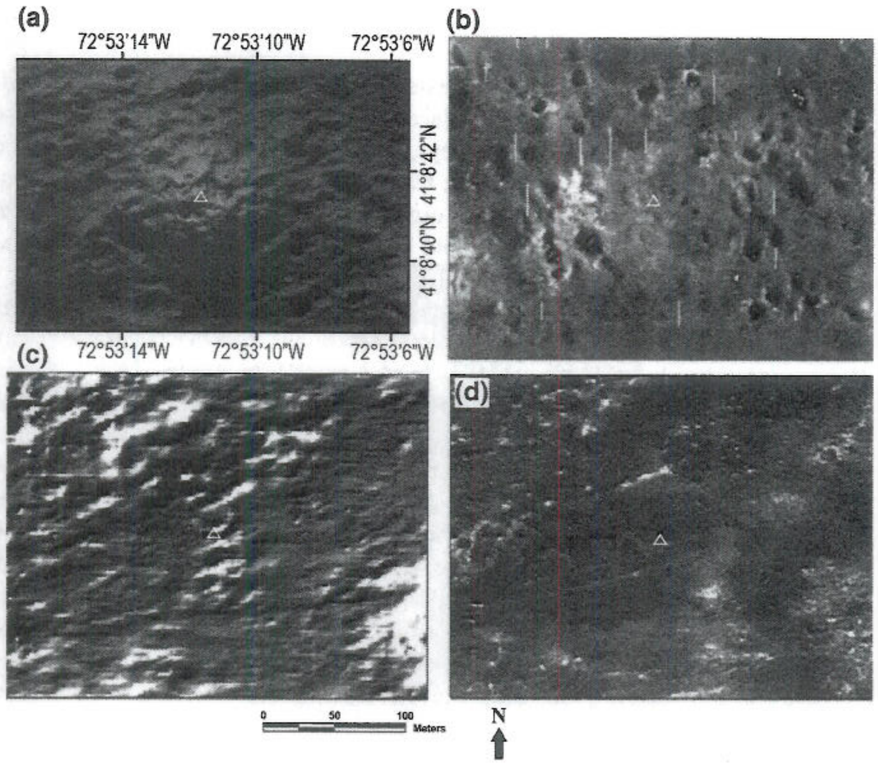

Fig. 6.34 Surface features within a dredged material disposal site in Long Island Sound before and after several disposal seasons. Images are centered over the location of a mound (CLIS-05) formed from October 2005 to May 2006. **a** Hillshaded multi-beam bathymetry from 2009, 3.5 years after mound formation. **b** Backscatter mosaic (sidescan imagery) from multibeam survey from 2009. **c** Hillshaded multi-beam bathymetry from 2005 prior to mound formation. **d** Sidescan sonar mosaic from 1997 (from Poppe et al. 2001). Individual ring features and impact craters are from single disposal events with split-hull disposal barges (from Valente et al. 2012; ENSR 2007)

of the physical processes described above, the benthic disturbance that results from placement of dredged material in open water habitats in LIS is a remobilization of surface sediments, burial of surface sediments and benthic infauna, and introduction of disturbed sediments with high organic loads into discrete areas (Germano et al. 1994). Virtually all benthic infauna are smothered in layers that exceed 15 cm. The ability to escape a given depth of burial is related to the life habits of the fauna (Kranz 1974; Maurer et al. 1986; Kjeilen-Eilertsen et al. 2004); strong burrowing deposit feeders can escape from 10 cm or more of burial (Jackson and James 1979; Bellchambers and Richardson 1995), but attached epifaunal suspension feeders cannot survive more than 1 cm (Kranz 1974). This means that some organisms can burrow up through thin layers of fresh sediment, but many will not. The center region of the disposal mounds formed in the Sound is usually sufficiently thick to catastrophically bury all organisms that cannot move quickly (Germano et al. 1994).

An "apron" of thinner material can extend 100–500 m beyond the bathymetrically detectable margin of the mound [accumulations greater than 10 cm can be reliably detected with multibeam surveys, greater than 20 cm with single beam fathometers (Fredette and French 2004; Carey et al. 2012)]. In the apron of thinner deposition, the introduction of high organic loads provides a surge in potential food supply for deposit feeders and rapid bioturbation usually obliterates the distinct layer within months (Germano et al. 2011).

The surface of the mound, including the apron, attracts high settlement densities of small surface deposit feeders (polychaetes, amphipods, bivalves, and meiofauna). This response has been documented by numerous monitoring studies in LIS and is consistent with the successional model of Rhoads et al. (1978) and Pearson and Rosenberg (1978). The nature and rate of recolonization can be strongly influenced by the timing of disturbance relative to seasonal pulses of settlement and growth of larvae (Zajac and Whitlatch 1982; Wilber et al. 2007). The successional model of response to physical disturbance from placement of dredged materials (Rhoads et al. 1978) has been tested with observation of disposal mounds in LIS since 1982 with the use of sediment profile imagery (Germano et al. 2011). Sediment profile imaging (SPI) utilizes a cross-sectional image of the upper 20 cm of the sediments to observe visual evidence of organism-sediment interactions. A phenomenological model (Rhoads and Germano 1982, 1986) has been used to interpret the ecological effects of dredged material in LIS (Germano et al. 1994) and minimize the impacts of disturbance (Fredette 1998; Fredette and French 2004).

The infaunal successional model (Rhoads and Germano 1986) posits that stage 1 organisms (small, tube-dwelling surface deposit feeders) appear within days or weeks of physical disturbance or deposition of a fresh layer of dredged material. If no further disturbance occurs, these stage 1 organisms are replaced by infaunal deposit feeders (stage 2) and eventually by larger infaunal deposit feeders (stage 3), many that feed in a head-down orientation that creates distinctive feeding voids (Germano et al. 2011). The establishment of this mature community may take months to years to complete and results in a deepening of the bioturbated mixed sediment layer and convergence with the surrounding benthic habitat conditions, depending on factors such as the spatio-temporal structure of the species pool (Zajac 2001). Potential variation in the rate of succession is illustrated by recent results collected from a disposal mound 5 months after cessation of disposal in 2009 (Fig. 6.35).

Benthic disturbance from dredged material disposal in LIS has immediate effects on sessile epifauna and infauna (Germano et al. 1994, 2011). The management approach to dredged material disposal in LIS includes biological testing of sediments and active management of disposal to segregate materials determined to be unsuitable for unconfined open water placement (Fredette and French 2004; Carey et al. 2006). During the development of the management approach, dredged material known to contain elevated levels of metals and PAHs was placed at the Central Long Island Sound Disposal Site in 1983 at several locations (capped and uncapped) as an experiment and monitored extensively

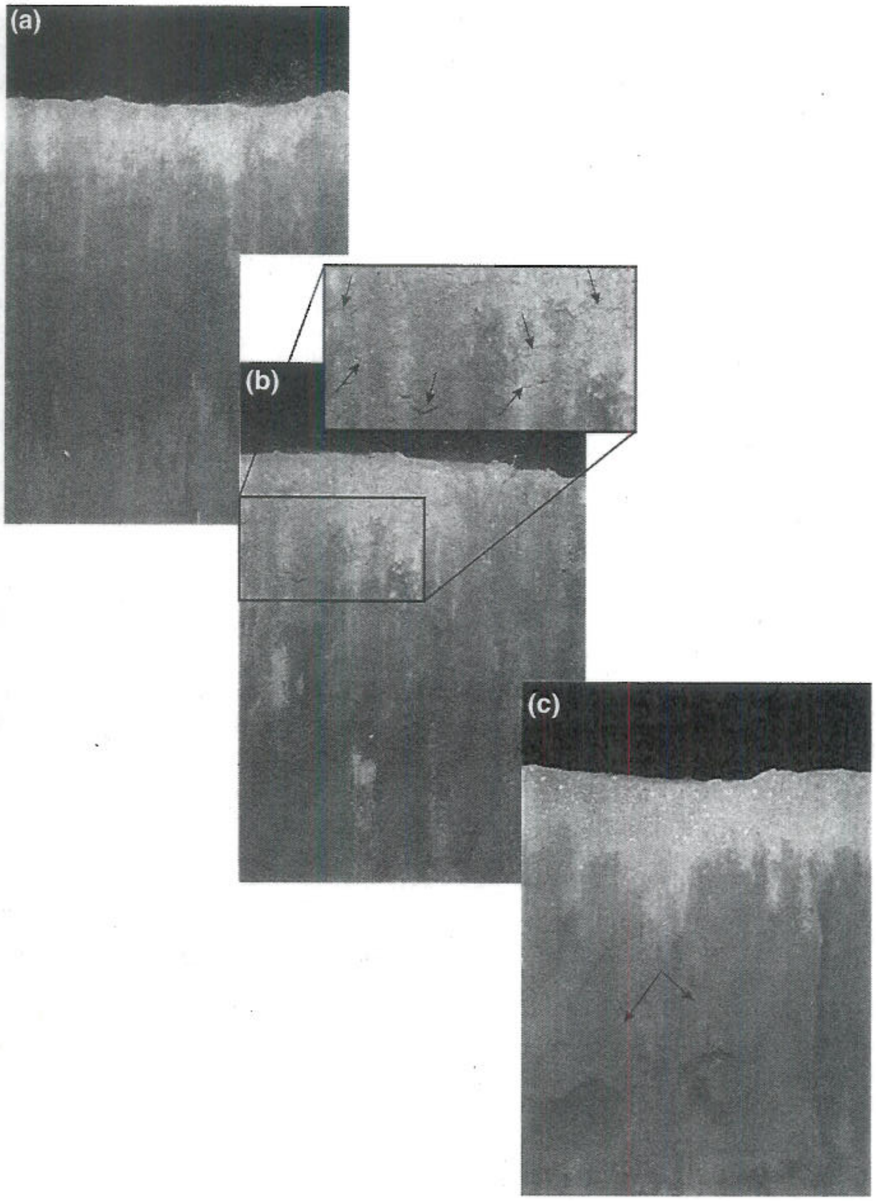

Fig. 6.35 Sediment profile images collected in October 2009 from a disposal mound (CLIS-08) formed in from October 2008 to May 2009 at the Central Long Island Sound Disposal Site. **a** Image illustrating dredged material colonized by tube-building worms (Stage 1). **b** Image illustrating a transitional successional status from Stage 1 to 2. Small Stage 1 worm tubes are visible at the sediment surface, and numerous small tunnels produced by burrowing Stage 2 meiofauna (e.g., crustaceans and bivalves) occur just below the surface (*arrows inset*). **c** Image showing a large vertical burrow and feeding voids (*arrows*) resulting in a Stage 3 successional designation (from Valente et al. 2012)

for more than 20 years (Germano and Rhoads 1984; Myre and Germano 2007). Short-term biological effects were observed after placement of unconfined dredged material (Myre and Germano 2007). This joint USEPA/USACE Field Verification Program was designed to field-verify existing test methods for predicting the environmental consequences of dredged material. The biological testing and resulting management approaches (sequestering dredged material with evidence of biological effects beneath a "cap" layer of material without significant biological effects) have contributed to the lack of observable long-term ecological effects from disposal activities in LIS. Apart from alterations of habitat due to introduction of different grain-size composition or changes in sediment transport conditions due to elevation of the sea floor, there is no evidence of long-term effects on benthic processes or habitat conditions (Germano et al. 2011).

6.6.2 Benthic Foraminifera

Foraminifera are unicellular, heterotrophic eukaryotes in the super-group Rhizaria, characterized by a branching, anastomizing network of granular reticulopodia (Adl et al. 2005). Many species have a proteinaceous theca, but many others make a shell (test) by agglutinating mineral grains in an organic or mineral matrix or by secreting $CaCO_3$. Tests may consist of one or many chambers. Foraminifera are marine, living from brackish coastal regions to the deepest ocean trenches (Pawlowski and Holzmann 2008).

There are about 50 species of living planktonic foraminifera and several thousand benthic ones (Murray 1991, 2006, 2007). Foraminifera are part of the meiobenthos, i.e., mostly between 63 and 1,000 μm in diameter. Benthic foraminifera are most diverse (hundreds of morphological species) along the lower continental shelf (Culver and Buzas 1982; Gooday 1993). Estuaries usually contain a few tens of species at most, coastal salt marshes and mangrove forests about 15 species, with 5–10 dominant species (Murray 1991, 2006; Scott et al. 2001; Javaux and Scott 2003).

Foraminifera are ubiquitous in the marine realm, their tests are easily fossilizable, and their small size makes it possible to obtain statistically valid data using relatively small samples (see e.g., Jorissen et al. 2007). Planktonic and benthic foraminiferal fossil assemblages thus have been used widely to reconstruct environmental changes on timescales from millions of years (see e.g., Thomas 2007) through historical times. In coastal regions, profound changes in foraminiferal fauna occurred partly in response to anthropogenic changes (see e.g., Alve 1995, 1996; Alve and Murray 1995; Culver and Buzas 1995; Karlsen et al. 2000; Scott et al. 2001; Platon et al. 2005, Murray 2006; Sen Gupta and Platon 2006; Nikulina et al. 2008; Gooday et al. 2009). In LIS, foraminifera have been used to study salt marsh ecology and reconstruct relative sea level rise, and eutrophication.